NeoBiota 66: 95–116 (2021)
doi: 10.3897/neobiota.66.67769
https://neobiota.pensoft.net

RESEARCH ARTICLE

A peer-reviewed open-access journal
NeoBiota
Advancing research on alien species and biological invasions

A weed risk analytical screen to assist in the prioritisation of an invasive flora for containment

F. Dane Panetta[1], Alasdair Grigg[2]

1 *Department of Agriculture and Food Systems, The University of Melbourne, Parkville, VIC 3010, Australia*
2 *Parks Australia, PO Box 867, Christmas Island, Indian Ocean 6798, Australia*

Corresponding author: F. Dane Panetta (dane.panetta@gmail.com)

Academic editor: S. Kumschick | Received 23 April 2021 | Accepted 17 June 2021 | Published 9 July 2021

Citation: Panetta FD, Grigg A (2021) A weed risk analytical screen to assist in the prioritisation of an invasive flora for containment. NeoBiota 66: 95–116. https://doi.org/10.3897/neobiota.66.67769

Abstract

Prioritising weeds for control and deciding upon the type of control and its associated investment are fundamental to weed management planning. Risk analysis is central to this process, combining the activities of risk assessment, risk management and risk communication. Risk assessment methodology has a rich history, but management feasibility has typically been a secondary matter, dealt with separately or not at all. Determinants of management feasibility for weeds include the stage of invasion, weed biology, means of control and cost of weed control. Here, we describe a simple weed risk analytical screen that combines risk assessment with species traits that influence management feasibility. We consider stage of invasion, species biological/dispersal characteristics and plant community invasibility in a preliminary analysis of the risk posed by the non-native plant species on Christmas Island in the Indian Ocean. For each of 31 high-risk species considered to be ineradicable under existing funding constraints, we analyse the risk posed to two major plant communities: evergreen closed-canopy rainforest and semi-deciduous scrub forest. Weed risk ratings are combined with ratings for species-intrinsic feasibility of containment (based on a measure that combines time to reproduction with potential for long distance dispersal) to create preliminary rankings for containment specific to each community. These rankings will provide a key input for a more thorough analysis of containment feasibility – one that considers spatial distributions/landscape features, management aspects and the social environment. We propose a general non-symmetric relationship between weed risk and management feasibility, considering risk to be the dominant component of risk analysis. Therefore, in this analysis species are ranked according to their intrinsic containment feasibility within similar levels of risk to produce an initial prioritisation list for containment. Shade-tolerant weeds are of particular concern for the closed-canopy evergreen rainforest on Christmas Island, but a greater diversity of weeds is likely to invade the semi-deciduous scrub forest because of higher light availability. Nevertheless, future invasion of both communities will likely be conditioned by disturbance, both natural and anthropogenic. The plant communities of Christmas Island have undergone significant

Copyright *F. Dane Panetta, Alasdair Grigg*. This is an open access article distributed under the terms of the Creative Commons Attribution License (CC BY 4.0), which permits unrestricted use, distribution, and reproduction in any medium, provided the original author and source are credited.

fragmentation because of clearing for phosphate mining and other purposes. With a substantial number of invasive plant species firmly established and having the potential to spread further, minimising future anthropogenic disturbance is paramount to reducing community invasibility and therefore conserving the island's unique biodiversity.

Keywords
Christmas Island, containment, dispersal, disturbance, eradication, invasibility, island ecosystems

Introduction

Risk analysis comprises three activities: risk assessment, risk management and risk communication (MacDiarmid and Pharo 2003; Vanderhoeven et al. 2017). Risk assessment, the evaluation of the threat (hazard) posed by a potentially invasive species, in conjunction with the likelihood of the threat being realised, has been employed to prioritise the management of invasive non-native species but generally fails to consider management feasibility (Booy et al. 2017; Bartz and Kowarik 2019). Bartz and Kowarik (2019) have recently reviewed the state of play for weed risk assessment in the broader context of the assessment of environmental impacts for harmful non-native plant species (hereafter generally referred to as 'weeds'). They recommened that assessments should incorporate context dependence of environmental impacts, as well as the prospects for successful management.

Weed risk management provides a structured evaluation of management options. It has received relatively little attention to date (Booy et al. 2017; Vanderhoeven et al. 2017; see also Bertolino et al. 2020 and Kumschick et al. 2020 for broader considerations of invasive organism risk management), but there has been a move by invasion scientists and practitioners to develop scoring protocols to assess the susceptibility of species to various management options (Booy 2015, cited in Vanderhoeven et al. 2017; Wilson et al. 2017). Among other considerations, these protocols are based on information relating to species distributions and abundance, the likelihood of reinvasion, the effectiveness of management options, relevant legislation and public acceptance of the management measures (Vanderhoeven et al. 2017).

Eradication as a weed management goal has been rather comprehensively investigated following seminal publications (Rejmánek and Pitcairn 2002; Panetta and Timmins 2004; Cacho et al. 2006). Wilson et al. (2017) provide a recent summary on the topic. The emerging conclusion is that eradication is feasible in a relatively small number of cases (Panetta 2015; Scott et al. 2019; Williams et al. 2019; Hulme 2020), owing largely to attributes that distinguish weeds from other invasive organisms, such as the potential for development of persistent seed banks. Difficulties in achieving eradication are generally more pronounced for weeds of natural, as opposed to highly modified ecosystems (Panetta and Timmins 2004; Panetta 2009; Moore et al. 2011).

Given the constraints on the applicability of eradication as a weed invasion management goal, containment emerges as a logical management alternative should

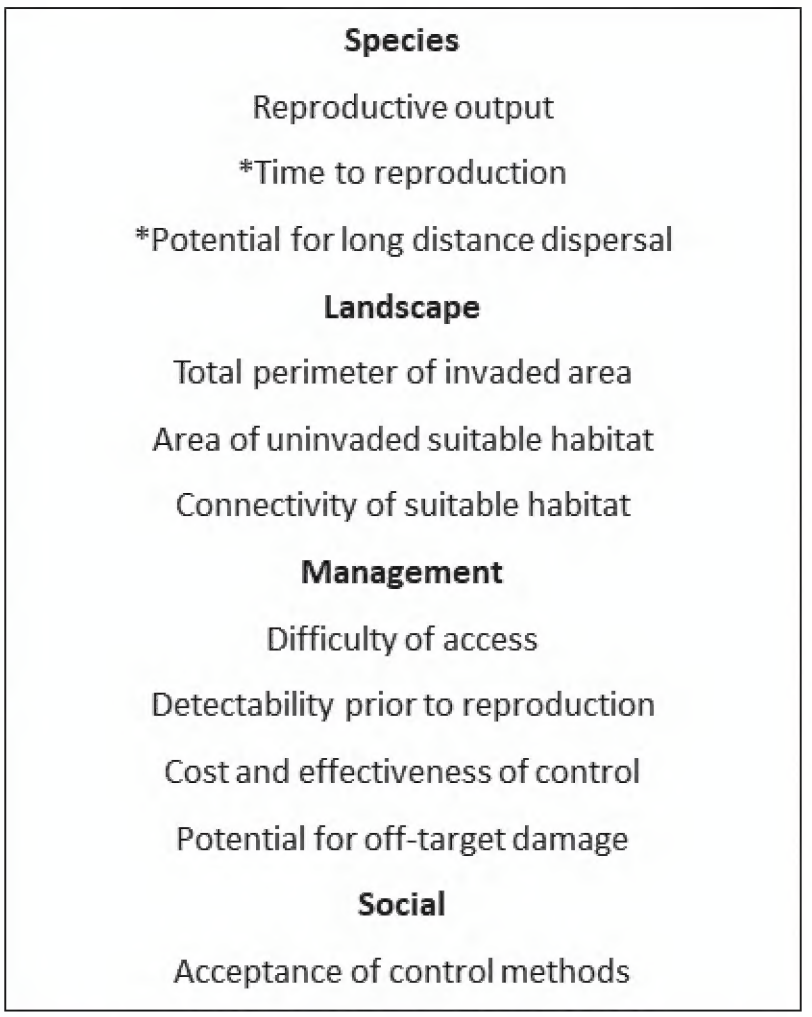

Figure 1. Key species traits and other aspects that affect containment feasibility (modified from Grice et al. 2010). Traits with asterisks are employed in this piece to obtain a preliminary ranking of weeds for containment.

eradication not be feasible (Panetta 2009; Wilson et al. 2011; but see Fletcher et al. 2015). Containment can be either absolute (stopping spread) or relative (slowing spread), but the concept of absolute containment has extremely limited application (Panetta and Cacho 2012), often restricted to a scenario combining species that naturally spread slowly with the existence of strong barriers (Hulme 2006). Slowing spread can provide substantial benefits (Sharov and Liebhold 1998), including 'buying time' while more effective control methods, such as biological control, are developed. However, this strategy requires an indefinite commitment of funding and other resources. This is perhaps why there has been relatively little research on containment as a weed management goal (Grice 2009; Grice et al. 2010; Panetta 2012; Panetta and Cacho 2012; Fletcher and Westcott 2013; Fletcher et al. 2015). Multiple factors need to be considered in the estimation of containment feasibility (Fig. 1).

Following the recommendations made by Bartz and Kowarik (2019), herein we consider the context dependence of species impacts, as well as the prospects for successful management. We develop a simple method that combines weed risk with species traits that influence their capacity to spread, with the aim of producing a ranking of species that could serve as a key input for a more detailed evaluation of containment feasibility. For an example of the application of this method we focus on the weeds that threaten the plant communities found on Christmas Island, a small and isolated island

located in the Indian Ocean. These species are well established on the island, but many still have restricted distributions. Present resourcing constraints mean that eradication of even one of these species is not an option (A. Grigg, pers obs.), hence our focus on containment as a management goal.

Methods

The study area

Christmas Island lies in equatorial waters in the Indian Ocean, 370 km south of Java Indonesia and 1400 km northwest of Australia. The island is a submarine volcanic seamount that has been above the ocean surface for approximately 5 million years (Ali and Aitchison 2020). Total land area is 135 km^2, most of which is greater than 150 m above sea level (ASL), forming a plateau; the highest point (Murray Hill) is 359 m ASL. The basalt core is mantled by a sequence of Eocene to Quaternary limestones that are often expressed as cliffs, scree slopes, outcrops and pinnacles (Barrie 1967; Grimes 2001). A few different soil types occur on the island, and they are predominantly derived from weathered substrate rocks (e.g. limestones or volcanics) or marine sediments accumulated during submergence phases. The depth of the soil profile is commonly less than 4 m but can range from zero (where limestone outcropping is present), to 80 m where fault or fissure infilling has occurred. Soils are often shallower on the geologically more recent lower terraces than on the central plateau.

The island has a tropical monsoonal climate, with most rainfall occurring between December and April. Long term average rainfall is 2200 mm pa, but this can be highly variable from year to year, ranging from 1067 mm (1987) to 5120 mm (2016) (Bureau of Meteorology 2020). Temperatures are very stable, with minimums of around 20 °C and maximums of around 29 °C. Relative humidity averages around 80%–85%. South-easterly trade winds dominate in the dry season (July to October), and north-westerly winds dominate in the summer wet season (Bureau of Meteorology 2020).

These conditions support a dense tropical rainforest across much of the island, with a canopy typically around 35 m and some emergent trees on the plateau approaching 50 m. The remote, isolated nature of this island has given rise to at least 253 unique species of fauna and flora, with endemic plants accounting for 17, possibly 18 species (Director of National Parks 2014; James et al. 2019). A total of 213 species of native plants has been reported (Claussen 2005). The Flora of Australia (1993) provides the most comprehensive description of vegetation types on the island, including seven old-growth/primary native vegetation types and two categories of secondary vegetation on historically cleared areas. Geoscience Australia (2014) developed a map showing vegetation and clearing patterns based on aerial images and LIDAR data (captured in 2011). A simplified form showing all five primary native vegetation types and historically cleared areas is shown in Fig. 2.

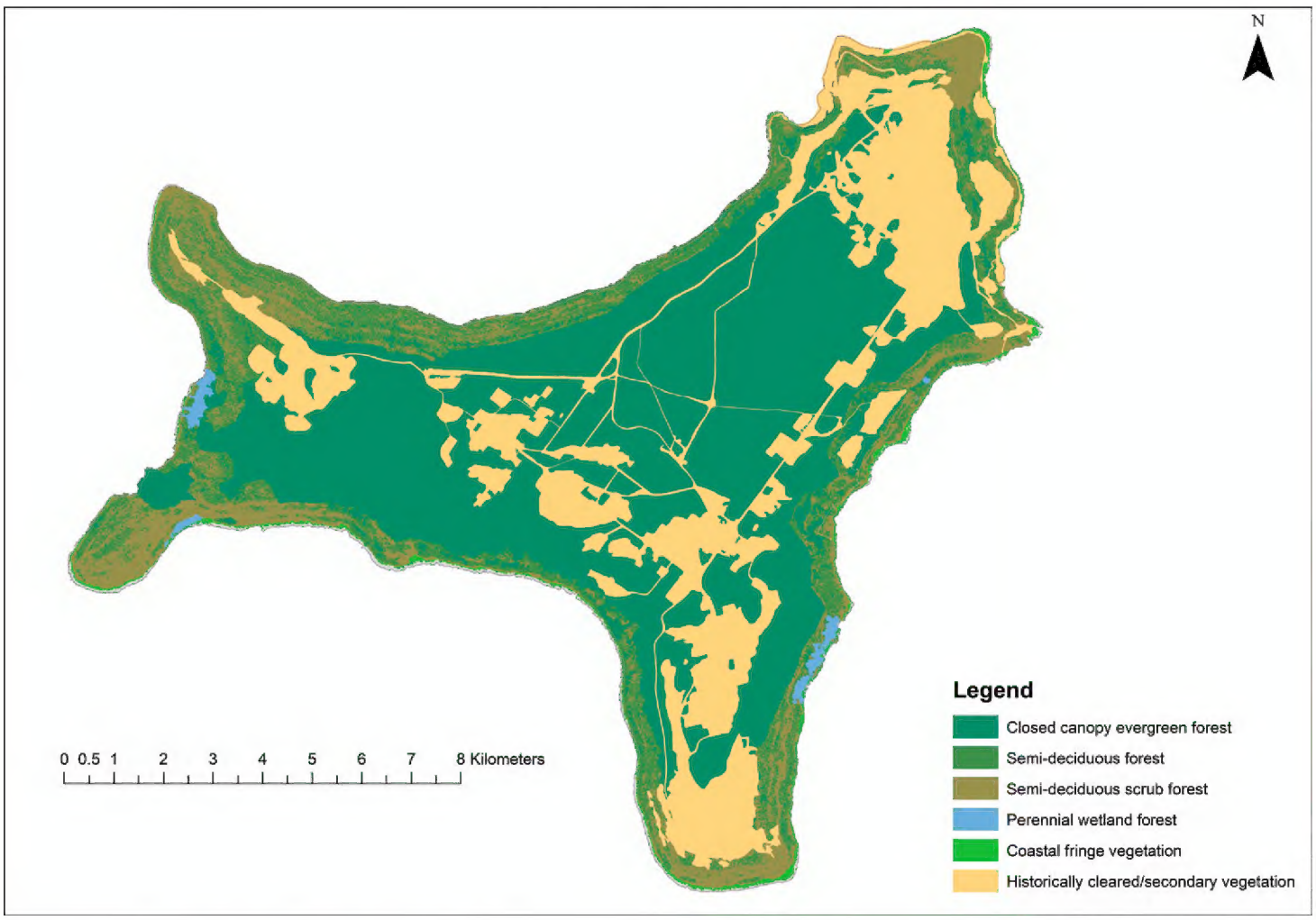

Figure 2. Main vegetation types and historical clearing boundary on Christmas Island (data courtesy of Geoscience Australia 2014).

Rainforest on Christmas Island is tallest over areas with deep soil (Flora of Australia 1993). In areas where limestone is nearer the surface (on the terraces), tree heights are reduced, the canopy is more open and diversity is often higher (Flora of Australia 1993). Forest on the plateau is evergreen with a closed canopy (>90% foliar cover) throughout the year. Vegetation on the terraced margins of the island is semi-deciduous (including both semi-deciduous forest and semi-deciduous scrub forest) and during long dry periods can become quite open (<50% foliar cover). This increases the invasibility of these forest types (Swarbrick and Hart 2000; Fine 2002; Green et al. 2004). Tree falls are common in all forest types, especially during wet season storms, and can provide opportunities for weed invasion.

The greatest disturbance factor, however, has been mechanised clearing for phosphate mining. Approximately 25% of the island's rainforest has been cleared for mining and associated developments since the island was settled in the 1880s (Fig. 2). As a quarry-type operation, large areas are cleared to remove the soil profile (~4 m), often leaving behind little or no soil on the pit floor. The post mining landscape is typically characterised by limestone pinnacles that can be deeply incised, sometimes standing over 5 m tall. Once equipment has dug out such areas, it can be impossible or too costly to access them again, meaning they can harbour weeds and be beyond control efforts. Only a small fraction of the total mined area can be rehabilitated sufficiently to restore even basic ecological functionality. Remediation works involve the use of heavy machinery, such as

dozers, excavators, loaders and haul trucks, to rip compacted pit floors, break down limestone pinnacles and bring in what little soil is available from backfill stockpiles, generally at great expense. Early attempts at rehabilitation by mining companies in the 1970s involved planting a range of non-native tree species such as *Aleurites moluccana*, *Melia azedarach*, *Sesbania grandiflora*, *Syzygium jambos* and *Spathodea campanulata*, many of which have persisted to become weeds (Carew-Reid 1987). Since 1992, rehabilitation efforts have been carried out by the Australian Government's Christmas Island Minesite to Forest Rehabilitation program, using funds paid by the current mining company as a condition of its lease. New and old areas are being rehabilitated, with around 20,000 native trees planted each year. Weed control is a major component of the program.

Edges between rainforest and cleared areas, such as along roads, former drill lines for mineral exploration, railway lines and around mining pits, are extensive (Swarbrick and Hart 2000). Approximately 63% of the island is national park, 93% of which is old-growth primary forest (7851 ha) and the remainder is mostly rehabilitation fields. In addition, approximately 2337 ha of old-growth, primary forest is currently held on Unallocated Crown Land. Including pockets elsewhere, there is a collective total of ~10,215 ha of old-growth primary vegetation (all naturally occurring vegetation types). These virgin forest areas host the island's endemic and formally listed threatened species, and therefore are the priority for managing impacts from invasive species and human disturbance.

Species-intrinsic weed containment feasibility

Panetta (2015) proposed a method for categorising plant species in terms of the species-intrinsic feasibility of eradication, based upon three attributes: 1) the time to reproduction (i.e., duration of the juvenile period); 2) the level of seed persistence; and 3) the potential for long distance dispersal (LDD), which has a disproportionate effect on the rate of spread (Nathan 2006).

In the case of containment feasibility, seed persistence is considered to be of secondary importance because the primary management focus is on the spread of a plant from a site rather than its persistence there (Wilson et al. 2017). The ability to contain a weed will depend therefore upon the potential for controlling its reproductive output, as well as the subsequent dispersal of whatever propagules are produced despite management actions (Panetta and Cacho 2014). Thus, the key determinants of containment feasibility are time to reproduction and the potential for LDD. A proxy for the latter is whether the dispersal suite (sensu Panetta 2012) is dominated by natural dispersal vectors such as wind, water or wild animals, or by humans and their agents. For the Christmas Island weeds under consideration, wind and wild animals (birds and bats) are the most active contributors to LDD (Swarbrick and Hart 2000). Potential reproductive output is another trait that can be considered in the determination of containment feasibility (Fig. 1). However, we do not address it here because a fundamental component of containment is the prevention of reproduction, even though some reproductive escape (Panetta 2007) may occur.

Weed risk assessment

Consistent with the definition of risk assessment as threat (or hazard) × likelihood (Daehler and Virtue 2010), weed risk assessment involves an evaluation of species in terms of both their impact (threat) and their invasiveness (i.e., potential for spread, sensu Richardson et al. 2000). Various flora surveys over the past century, including several conducted during the last few years (by the Australian National Herbarium 2012 and 2019; Department of Agriculture, Water and Environment 2019; and Parks Australia), plus a review by Lohr et al. (2016), have identified more than 600 introduced species on Christmas Island. From this pool of introduced species, the whole-of-island weed risk was assessed for 130 species that had shown evidence of naturalisation. Personnel (n = 10) from Parks Australia, the National Herbarium, the Department of Agriculture, Water and Environment, and private consultants provided input that was compiled by Parks Australia to reach consensus. Many of the factors included in the Australian weed risk assessment model (Pheloung et al. 1999), such as potential adaptation to local conditions, weed history, plant type, undesirable traits, reproduction, dispersal mechanisms and persistence attributes were considered. More than 70 species were deemed problematic for secondary vegetation types, such as rehabilitation plots and other disturbed areas. Given that our focus here relates only to old-growth, primary vegetation types, species whose impact was restricted to secondary vegetation types were removed. This resulted in a pool of 31 priority species (Table 1) that had been assessed as 'high', 'very high' or 'extreme' risk, a ternary categorisation resembling that employed by Booy et al. (2017) to prioritise non-native organisms for eradication.

From these whole-of-island risk assessments, species were scored according to the risk posed to two major plant communities on the island: closed-canopy evergreen rainforest (on the deep soils of the island's central plateau) and semi-deciduous scrub forest (on the shallow soils of the coastal terraces). Among the natural, primary vegetation communities on the island, these two vegetation types represent the endpoints of a continuum of invasibility, with the former being less invasible than the latter and semi-deciduous forest representing a transition between the two (Swarbrick and Hart 2000). In general, pristine tropical rainforests have a low invasibility for most weeds, and only shade tolerant non-native species pose a threat (Fine 2002) – that is, until a disturbance (natural or anthropogenic) takes place and allows opportunities for invasion by a broader range of weed species. The exclusion of most weeds by tropical rainforests has been attributed to the fact that the great majority of species that are transported to such areas lack specific life history traits, most importantly shade tolerance, necessary for invasion in the absence of disturbance (Fine 2002).

Species designated 'extreme risk' were those considered to have the potential to transform (Richardson et al. 2000) at least one of the communities. These species were scored as '4'. Those posing a greater risk to one community than the other were designated as 'very high' risk (scored as '3') in the former and as 'high' risk (scored as '2') in the latter. In some cases, low recruitment, survival and growth had been observed in one of the communities (e.g., *Syzygium grande* and *Hevea brasiliensis* in semi-deciduous

Table 1. Weed risk assessment (WRA) and containment feasibility (CF) ratings for the 31 species considered to pose the greatest threat to two major plant communities on Christmas Island. WRA ratings are 4 (extreme); 3 (very high); 2 (high); and 1 (other). CF ratings are based on species attributes of juvenile period (scored as 1 for < 2 yr and 2 for > 2 yr) and potential for long distance dispersal (scored as 1 for species that are wind- or bird- and bat-dispersed and 2 for those whose dispersal occurs primarily through gravity or explosive dehiscence).

Growth form	Species	Risk assessment rating		Containment feasibility rating		
		Closed-canopy evergreen rainforest	Semi-deciduous scrub forest	Juvenile period	Dispersal	Total
Tree	*Adenanthera pavonia*	2	2	2	2	4
Tree	*Aleurites moluccana*	2	2	2	2	4
Tree	*Castilla elastica*	3	1	2	1	3
Tree	*Clausena excavata*	4	3	2	1	3
Tree	*Delonix regia*	2	2	2	2	4
Tree	*Ficus elastica*	2	1	2	1	3
Tree	*Hevea brasiliensis*	2	1	2	2	4
Tree	*Jatropha curcas*	2	3	2	2	4
Tree	*Leucaena leucocephala*	2	4	1	2	3
Tree	*Manihot glaziovii*	2	2	2	2	4
Tree	*Melia azederach*	3	3	2	1	3
Tree	*Piper aduncum*	2	3	2	1	3
Tree	*Pithecellobium dulce*	2	3	2	1	3
Tree	*Psidium cattleyanum*	2	3	2	1	3
Tree	*Psidium guajava*	2	3	2	1	3
Tree	*Spathodea campanulata*	3	3	2	1	3
Tree	*Syzygium grande*	3	1	2	1	3
Shrub	*Pluchea indica*	2	3	1	1	2
Shrub	*Tecoma stans*	1	3	1	1	2
Vine	*Antigonon leptopus*	2	4	1	2	3
Vine	*Calopogonium mucunoides*	2	3	1	2	3
Vine	*Centrosema pubescens*	1	3	1	2	3
Vine	*Ipomoea cairica*	1	3	1	2	3
Vine	*Ipomoea nil*	1	3	1	2	3
Vine	*Macroptilium atropurpureum*	1	3	1	2	3
Vine	*Mikania micrantha*	3	4	1	1	2
Vine	*Mucuna albertisii*	3	2	2	2	4
Vine	*Mucuna gigantea*	3	2	2	2	4
Vine	*Mucuna pruriens*	3	2	2	2	4
Vine	*Paederia foetida*	2	3	1	1	2
Vine	*Tinospora crispa*	2	3	1	2	3

scrub forest and *Tecoma stans*, *Ipomoea* spp. and *Macroptilium atropurpureum* in the closed evergreen forest). These species were designated 'other' risk (scored as '1'), such that all lower categories of risk were combined.

Chromolaena odorata (Siam weed) is another species that could potentially transform semi-deciduous forest types. A small, but seed-producing population was discovered in 2010 (Dodd et al. 2012) and was immediately destroyed. Recruits, all of which were controlled, continued to emerge during wet seasons for seven years. Monitoring is continuing – extensive searches for this species by Parks Australia staff have been conducted at least twice per year, but it has not been detected in any other location. While

exhaustion of the *C. odorata* seed bank at the known outbreak site has likely occurred, there remains a possibility that this weed remains undetected elsewhere, or could re-invade the island, most likely from the Cocos Keeling Islands (~970 km away) via sea or air transport. Should *C. odorata* be detected again on Christmas Island, it would be targeted for eradication so it is not considered further.

We reiterate that numerous species were excluded from the priority list because they are considered less of a threat to intact, virgin forest types, but following significant disturbance would possibly need to be brought back into consideration.

Combining risk assessment with species-intrinsic containment feasibility

As in the system proposed by Panetta (2015), species were classified according to whether their juvenile periods were less than or equal to 2 years, or more than 2 years (scored as '1' and '2' respectively – the higher score aligned with greater containment feasibility). Species whose potential for LDD is high were scored as '1' and those for which LDD potential is relatively low were scored as '2', again with the higher score aligned with greater containment feasibility. Containment feasibility score totals therefore ranged between 2 and 4 (Table 1). Because our emphasis was on the characteristics of species that contribute to their spread potential, we did not include plant community invasibility in our estimates of weed containment feasibility.

Priorities for containment were obtained for individual communities by combining weed risk assessment ratings with the containment feasibility ratings for each species.

Results

Four species (*Antigonon leptopus*, *Clausena excavata*, *Leucaena leucocephala* and *Mikania micrantha*) were categorised as posing extreme risk (risk assessment ratings of '4'), (Table 1). *M. micrantha* is relatively widely established across the island in cleared areas, forest rehabilitation fields, vehicle tracks and some walking tracks. It invades forest edges and gaps, posing a threat to all plant communities, and will likely invade future gaps since it is wind-dispersed and develops persistent seed banks (Brooks et al. 2008; Macanawai et al. 2018). *C. excavata* is a well-dispersed species that establishes in all forest types. It appears to be the most shade tolerant of the priority weeds identified here and can attain high densities within intact closed-canopy evergreen rainforest. What makes this species even more problematic is that a large proportion of its seedlings escapes the intense predation by land crabs, to which seedlings of most species, both native and non-native, are prone (Green et al. 1999; Green et al. 2004). *A. leptopus* and *L. leucocephala* have major impacts in the semi-deciduous scrub forest – the first as a smothering vine limiting natural recruitment and the second by developing monocultures, in part due to its allelopathic leaf litter (Chou and Kuo 1986; Ahmed et al. 2008).

The distributions of weed assessment ratings for the two communities were different, as more species posed either 'extreme' risk or 'very high' risk to the semi-deciduous

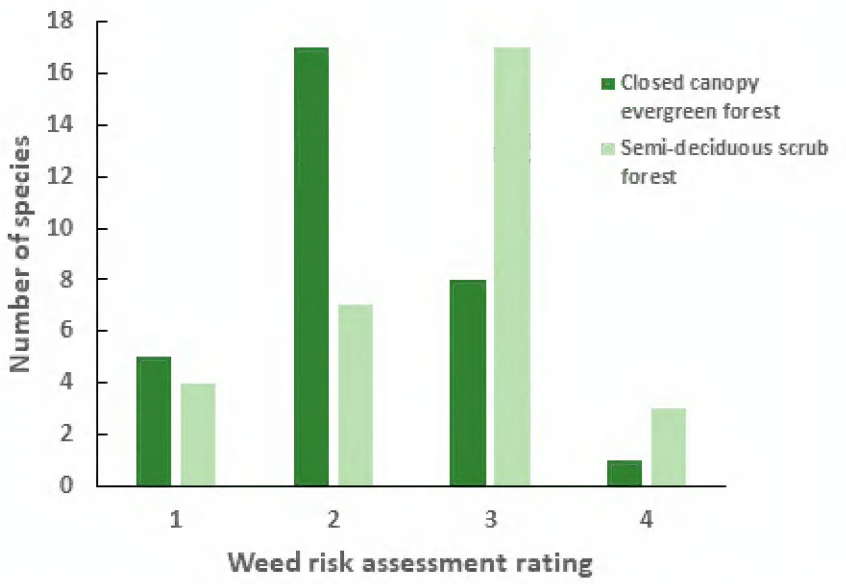

Figure 3. Distribution of weed risk assessment ratings for 31 species posing the highest threat to major plant communities on Christmas Island. WRA ratings are 4 (extreme); 3 (very high); 2 (high); and 1 (other).

scrub forest than to the closed-canopy evergreen forest (Fig. 3). These differences reflect the fact that invasion is dependent upon disturbances that reduce canopy cover in the closed-canopy evergreen rainforest, whereas semi-deciduous scrub forest has a higher base level of invasibility because it is more open.

Priorities for containment are shown for potentially invasive species in the closed canopy evergreen rainforest (Fig. 4) and semi-deciduous scrub forest (Fig. 5). For both communities there is a substantial number of tied values in the listed priorities.

Discussion

Virtue et al. (2001) designated the essential criteria for addressing the feasibility of controlling weeds as: 1) stage of invasion; 2) weed biology; 3) means of control; 4) cost of weed control; and 5) motivation of land managers. The management goal may be to eradicate a weed or to contain it spatially – sometimes collectively designated as 'co-ordinated control' (FAO 2011) – or it may be to maintain the abundance of a weed below an impact threshold (i.e., 'maintenance control'; Panetta and Gooden 2017). Different methods (e.g., Panetta and Timmins 2004; Booy et al. 2017) have been developed for evaluating eradication feasibility. These consider biological and ecological, socio-political, economic and operational factors specific to the context of an incursion.

The technical feasibility of co-ordinated control concerns the biological features and environmental context that, taken together, have a large effect on both the cost of control and the probability of management success (Cacho et al. 2006; Cacho and Pheloung 2007). The categorisation of species based upon the duration of the juvenile period, seed persistence, and dispersal characteristics (Panetta 2015) has provided a simple method for a first-pass determination of the species-intrinsic feasibility of eradication for a range of plant species (Panetta 2015). However, little guidance exists

Risk	Intrinsic containment feasibility	Species	Preliminary ranking
4	3	*Clausena excavata*	1
	2	*Mikania micrantha*	2
3	4	*Mucuna albertisii*	3
		Mucuna gigantea	3
		Mucuna pruriens	3
	3	*Castilla elastica*	4
		Melia azederach	4
		Spathodea campanulata	4
		Syzygium grande	4
2	4	*Adenanthera pavonia*	5
		Aleurites moluccana	5
		Delonix regia	5
		Hevea brasiliensis	5
		Manihot glaziovii	5
	3	*Antigonon leptopus*	6
		Calopogonium mucunoides	6
		Ficus elastica	6
		Jatropha curcas	6
		Leucaena leucocephala	6
		Piper aduncum	6
		Pithecellobium dulce	6
		Tinospora crispa	6
	2	*Paederia foetida*	7
		Pluchea indica	7
		Psidium cattleyanum	7
		Psidium guajava	7
1	3	*Centrosema pubescens*	8
		Ipomea cairica	8
		Ipomea nil	8
		Macroptilium atropurpureum	8
	2	*Tecoma stans*	9

Figure 4. Prioritisation for containment of weeds that are potentially invasive in the closed-canopy evergreen forest.

Risk	Intrinsic containment feasibility	Species	Preliminary ranking
4	3	*Antigonon leptopus*	1
		Leucaena leucocephala	1
	2	*Mikania micrantha*	2
3	4	*Centrosema pubescens*	3
		Clausena excavata	3
		Ipomea cairica	3
		Ipomea nil	3
		Macroptilium atropurpureum	3
		Piper aduncum	3
	3	*Calopogonium mucunoides*	4
		Jatropha curcas	4
		Melia azederach	4
		Pithecellobium dulce	4
		Spathodea campanulata	4
		Tecoma stans	4
		Tinospora crispa	4
	2	*Paederia foetida*	5
		Pluchea indica	5
		Psidium cattleyanum	5
		Psidium guajava	5
2	4	*Adenanthera pavonia*	6
		Aleurites moluccana	6
		Delonix regia	6
		Manihot glaziovii	6
		Mucuna albertisii	6
		Mucuna gigantea	6
		Mucuna pruriens	6
1	4	*Hevea brasiliensis*	7
	3	*Castilla elastica*	8
		Ficus elastica	8
		Syzygium grande	8

Figure 5. Prioritisation for containment of weeds that are potentially invasive in the semi-deciduous scrub forest.

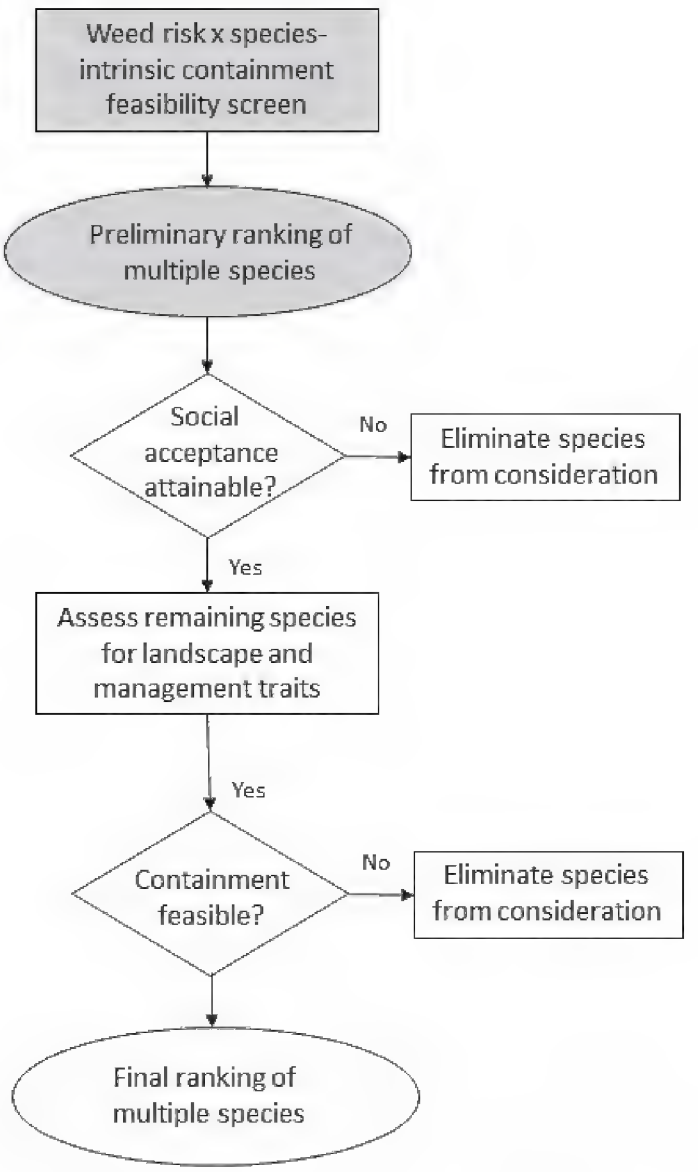

Figure 6. Contribution of initial screening to weed risk analysis. Grey shaded elements are addressed in the present treatment. Note that some species may need to be eliminated from consideration owing to social considerations.

in relation to the estimation of containment feasibility (Grice 2009; Grice et al. 2010; Panetta and Cacho 2012; Wilson et al. 2017).

In the present exercise we have modified the eradication algorithm of Panetta (2015) to this end. The duration of the juvenile period determines the return time for control (Panetta 2007) and the dispersal mechanisms in play determine how readily new foci of infestation might originate from plants whose reproductive output is reduced but not entirely prevented, or plants that escape detection altogether (Panetta and Cacho 2012). The inclusion of actual juvenile periods of weeds on Christmas Island would likely reduce the number of ties in the current species ranking.

Place of preliminary ranking in the estimation of weed containment feasibility

Consideration of the spatial distribution of each species, landscape features, management aspects and social considerations will be required for the more thorough weed risk analytical assessment. Our weed risk analytical screen is a key input to the analytical process (Fig. 6) because it encapsulates the risk posed by each species, as well as its

potential for spread. The latter is largely determined by how quickly a species produces propagules, in conjunction with propagule adaptations that increase the likelihood of long distance dispersal.

Exploring the relationship between weed risk and management feasibility

In our view the relative importance of these components of weed risk analysis requires further investigation. In prioritising species for eradication Booy et al. (2017) assumed a symmetric relationship, i.e., a species that was evaluated as posing a high risk in conjunction with medium feasibility of management should receive the same priority as one that posed a medium risk and high management feasibility. Taken to the extreme, the acceptance of this relationship means that a low risk species with high management feasibility would be accorded the same priority as a species manifesting the opposite combination, but we question the rationale of allocating scarce resources to the management of any species considered to pose a low risk. We maintain, on the other hand, that risk should have a higher weighting – given resourcing constraints, only the species that pose the highest risk should be considered for coordinated control, with prioritisation within this group informed by management feasibility (as shown in Figs 4 and 5 for weed risk analytical screening). We consider this assertion to have general application, not being restricted to the system under examination. A consequence of lower technical feasibility is that more resources will be required to achieve a given management goal (Panetta 2015; Wilson et al. 2017). In the present context, extreme-risk species may still warrant containment even if projected management costs are high relative to those associated with species posing lower risk. This is a prime example of the application of the precautionary principle in risk management, as advocated recently by Strubbe et al. (2019).

Setting subgoals for containment – the risk of false precision

Two fundamental issues must be considered when attempting to set subgoals for a containment program. The first relates to the counterfactual scenario (i.e., the rate of spread that would occur in the absence of intervention) against which the degree of spread reduction could be assessed. The second relates to the value of reducing this baseline spread rate.

Various models have been used to estimate spread rates of different organisms, including non-native plants (Higgins and Richardson 1996; Parry et al. 2013). Each modelling approach has its strengths and weaknesses, but key concerns include scale and complexity, the choice of theoretical versus empirically based models and, particularly relevant to the spread of plants, the existence of multiple dispersal pathways. Estimations of model and parameter uncertainty will be important for the purposes of decision making (Parry et al. 2013). It is likely that any modelled baseline spread for an invasive plant would be accompanied by wide confidence intervals, especially where the potential for long distance dispersal is a critical factor.

As was stated at the outset, slowing spread can provide substantial benefits. Net benefits of containment, which incorporate the costs of containment actions, can be identified through bioeconomics, i.e., the optimal management of renewable biological resources. Bioeconomic models can be used when only rough estimates of benefits and costs are available (Cacho 2006). However, any form of benefit/cost analysis of invasive species management is subject to biases that may lead to underestimation of costs and overestimation of benefits (Dahlsten et al. 1989; Myers et al. 1998) and therefore can be associated with "extremely wide confidence intervals" (Simberloff 2003, p. 251). The net value of a containment program (benefits accrued minus program costs) can be expected to vary with the degree of reduction of spread. The theoretical form of this relationship is a matter for speculation, but it is possibly non-linear, with increasing net benefits per unit spread reduction as absolute containment is approached.

Since the reduction of spread and its value can be expected to have wide confidence limits, it is important to be aware of the risk of false precision (Roberts 2017). In the first instance, the articulation of precise management goals (such as "70% reduction of spread") should be avoided. In the second, the degree of confidence in a measure such as the unit risk reduction (in terms of damages averted per dollar invested) might not be high, however valid such quantification might be from a theoretical perspective. Rather, given an acceptable estimate of containment feasibility, we suggest the way forward may be to adopt containment as an 'aspirational' objective and use measurements of spread over time (Panetta 2012) as a basis for deciding whether to continue a containment program or switch to an alternative strategy, such as asset protection (Auld and Johnson 2014). This is analogous to the approach that is taken with weed eradication, which is also essentially an aspirational goal. If over time an eradication program fails to demonstrate progress, this management goal may be abandoned (Panetta 2015). The goal of absolute containment has a discrete set of potential outcomes (spread is either stopped or it is not). For relative containment, however, a multitude of outcomes would meet the management objective – strictly speaking, any non-zero degree of spread reduction less than absolute containment. The determination of what is an acceptable level of spread reduction is clearly beyond the scope of this paper. As per the argument presented above, the output of any such determination is likely to be a range of values, the breadth of which would be related to levels of uncertainty in the counterfactual scenarios of weed spread rate and impact in the absence of management intervention.

Reducing invasibility by managing disturbance

The closed-canopy evergreen rainforest is intrinsically less invasible in the absence of disturbance, but at the same time hosts most of the island's endemic and formally listed threatened plant species. *Clausena excavata* is a rare example of a weed that can establish under an intact canopy and should be prioritised for control, whether via a formal containment program or in terms of asset protection. It has proven to be an invasive non-native weed in tropical forests elsewhere around the world (Viera et al. 2010; Roseleine and Suzuki 2012; Biswas and Das 2014). Growing slowly under low light

conditions, the species probably depends upon gaps created by tree falls to become reproductive. Seven other species are considered to pose a very high, and 17 species a high risk to the rainforest (Fig. 4).

Tropical cyclones have been relatively uncommon for Christmas Island, with roughly only one significant system impacting the island every decade or so. However, if cyclones become more intense and/or more frequent because of climate change, this could have major implications for the invasibility of the island's primary vegetation types, including closed-canopy evergreen forest. A cyclone in 2014 (TC Gillian) caused significant damage across the island, stripped much of the forest canopy and gave *M. micrantha* the opportunity to greatly expand its range into intact forest areas where it had never previously been able to establish. Given that this species has seed that can persist in the seedbank for up to seven years (Brooks et al. 2008; Macanawai et al. 2018), it will likely become more problematic if cyclones occur more frequently. That said, an effective biological control agent, the rust fungus *Puccinia spegazzinii* (Day and Riding 2019) has now been approved for release on the Australian mainland and may soon be available for use on Christmas Island, pending the outcome of a separate import approval process. When widely established, the rust should reduce the effort required to control *M. micrantha* by other means. This illustrates the containment benefit of 'buying time' until a more sustainable control measure becomes available.

The native forests of Christmas Island have been substantially cleared and fragmented by mining for phosphate minerals. Island ecosystems and island endemic species are notoriously vulnerable to impacts, declines and extinctions because of habitat disturbance and the associated compounding threats of invasive flora and fauna (D'Antonio and Dudley 1994; Lonsdale 1999; Denslow 2003; Gimeno et al. 2006; Vilà et al. 2006; Reaser et al. 2007; Baider and Florens 2011; Andrew et al. 2018; Heger and Andel 2019; Brock and Daehler 2020). In addition to more than 3,000 hectares of broad-scale clearing, approximately 500 km of mineral exploration drill lines have also been made in a systematic grid pattern across much of the island, predominantly through the closed-canopy evergreen forest on the plateau (Parks Australia 2009). Most of these drill lines were pushed through in the 1950s, 60s and 70s, before many of the weed species of current concern were widely distributed on the island. In the relative absence of problem weeds, drill lines once naturally regenerated by successional processes involving native species. The vegetation on old drill lines is often almost indistinguishable from primary forest. From an ecological perspective it has regained full ecosystem functionality, with mature trees offering bird nesting opportunities and an abundance of red crab burrows. The expansion of numerous weed species across the island in recent decades has significantly altered the ecological dynamics of colonisation and succession. Aggressive invaders such as *C. excavata*, *M. micrantha*, *L. leucocephala* and others now overgrow and out-compete native recruits and have been observed colonising recent drill lines and cleared areas. There are now many weeds that will colonise and dominate larger disturbed areas (Fig. 4). Furthermore, newly formed edges to intact vegetation enable weeds – especially the more shade tolerant species such as *C. excavata* – to encroach into primary vegetation. Left unchecked, there is little doubt

that weeds will dominate post clearing recruitment, cause major edge effects and progressively invade forest types on the island. This highlights the critical importance of minimising further forest fragmentation via all forms of anthropogenic disturbance as part of a holistic approach to weed risk management.

Conclusion

Decisions concerning which weeds of natural ecosystems are targeted for control, as well as the type of control that is to be undertaken, need to be based on the degree of risk posed and the degree of difficulty that can be anticipated in attempting to manage it. Prioritisations of species for control need to be framed with reference to individual plant communities, since weed risk can be expected to vary according to the environmental context. Stringent limitations in the availability of resources for management generally imply that only the species that pose the highest levels of risk should be considered for co-ordinated control. We have shown that traits that contribute to the species-intrinsic feasibility of eradication may also be useful in assessing the feasibility of containment for species with restricted distributions, thereby assisting in the prioritisation of weeds to this management end. The development of a method for considering the spatial distribution of each species, landscape features, management aspects and social considerations is underway.

Acknowledgements

We thank Parks Australia, particularly the Christmas Island National Park Manager, Kerrie Bennison, for inviting the senior author to visit the island and supporting the cost of flights and accommodation. We also thank CINP staff for their assistance with fieldwork, particularly Trent Lane and Roslan Sani. Nick MacGregor and two anonymous reviewers provided valuable feedback on earlier drafts.

References

Ahmed R, Hoque AR, Hossain MK (2008) Allelopathic effects of *Leucaena leucocephala* leaf litter on some forest and agricultural crops grown in nursery. Journal of Forestry Research 19: 298–302. https://doi.org/10.1007/s11676-008-0053-0

Ali JR, Aitchison JC (2020) Time of re-emergence of Christmas Island and its biogeographical significance. Palaeogeography, Palaeoclimatology, Palaeoecology 537: e109396. https://doi.org/10.1016/j.palaeo.2019.109396

Andrew P, Cogger H, Driscoll D, Flakus S, Harlow P, Maple D, Misso M, Pink C, Retallick K, Rose K, Tiernan B (2018) Somewhat saved: a captive breeding programme for two endemic Christmas Island lizard species, now extinct in the wild. Oryx 52: 171–174. https://doi.org/10.1017/S0030605316001071

Auld B, Johnson SB (2014) Invasive alien plant management. CAB Reviews 9: 1–12. https://doi.org/10.1079/PAVSNNR20149037

Baider C, Florens FV (2011) Control of invasive alien weeds averts imminent plant extinction. Biological Invasions 13: 2641–2646. https://doi.org/10.1007/s10530-011-9980-3

Barrie J (1967) The Geology of Christmas Island – Part 1. Bureau of Mineral Resources, Geology and Geophysics. Commonwealth of Australia, Canberra.

Bartz R, Kowarik K (2019) Assessing the environmental impacts of invasive alien plants: a review of assessment approaches. NeoBiota 43: 66–99. https://doi.org/10.3897/neobiota.43.30122

Bertolino S, Ancillotto L, Bartolommei P, Benassi G, Capizzi D, Gasperini S, Lucchesi M, Mori E, Scillitani L, Sozio G, Falaschi M, Ficetola GF, Cerri J, Genovesi P, Carnevali L, Loy A, Monaco A (2020) A framework for prioritising present and potentially invasive mammal species for a national list. NeoBiota 62: 31–54. https://doi.org/10.3897/neobiota.62.52934

Biswas K, Das AP (2014) Impact of invasive alien weeds on phytodiversity of Terai–Duars region of West Bengal, India. NBU Journal of Plant Sciences 11: 15–31.

Booy O (2015) Risk management and prioritisation in GB. GB Non-native Species Secretariat.

Booy O, Mill AC, Roy HE, Hiley A, Moore N, Robertson P, Baker S, Brazier M, Bue M, Bullock R, Campbell S, Eyre D, Foster J, Hatton-Ellis M, Long J, Macadam C, Morrison-Bell C, Mumford J, Newman J, Parrott D, Payne R, Renals T, Rodgers E, Spencer M, Stebbing P, Sutton-Croft M, Walker KJ, Ward A, Whittaker S, Wyn G (2017) Risk management to prioritise the eradication of new and emerging invasive non-native species. Biological Invasions 19: 2401–2417. https://doi.org/10.1007/s10530-017-1451-z

Brock KC, Daehler CC (2020) Applying an invasion and risk framework to track non-native island floras: a case study of challenges and solutions in Hawai'i. NeoBiota 62: 55–79. https://doi.org/10.3897/neobiota.62.52764

Brooks SJ, Panetta FD, Galway KE (2008) Progress towards the eradication of mikania vine (*Mikania micrantha*) and limnocharis (*Limnocharis flava*) in northern Australia. Invasive Plant Science and Management 1: 296–303. https://doi.org/10.1614/IPSM-08-067.1

Bureau of Meteorology (2020) Climate statistics for Australian locations. Monthly climate statistics. Christmas Island Aero. http://www.bom.gov.au/climate/averages/tables/cw_200790_All.shtml

Cacho OJ (2006) Bioeconomics of invasive species in aquatic ecosystems. Aquaculture Economics and Management 10: 107–124. https://doi.org/10.1080/13657300600695616

Cacho OJ, Pheloung P (2007) WeedSearch: Weed eradication feasibility analysis, Software manual. Co-operative Research Centre for Australian Weed Management, Project 1.2.8, August 2007, University of New England, Australia. https://www.une.edu.au/__data/assets/file/0006/103668/Cacho_Software.zip

Cacho O, Spring D, Pheloung P, Hester S (2006) Evaluating the feasibility of eradicating an invasion. Biological Invasions 8: 903–917. https://doi.org/10.1007/s10530-005-4733-9

Carew-Reid J (1987) Rehabilitation of mined clearings on Christmas Island, Indian Ocean. Australian National Parks and Wildlife Service, Canberra.

Chou CH, Kuo YL (1986) Allelopathic research of subtropical vegetation in Taiwan. Journal of Chemical Ecology 12: 1431–1448. https://doi.org/10.1007/BF01012362

Claussen J (2005) Native plants of Christmas Island. Australian Biological Resources Study, Canberra.

Daehler CC, Virtue JG (2010) Likelihood and consequence: reframing the Australian Weed Risk Assessment to reflect a standard model of risk. Plant Protection Quarterly 25: 51–55.

Dahlsten DL, Garcia R, Lorraine H (1989) Eradication as a pest management tool: concepts and contexts. In: Dahlsten DL, Garcia R, Lorraine H (Eds) Eradication of exotic pests. Analysis with case histories. Yale University Press, New Haven, CT, 3–15. https://doi.org/10.2307/j.ctt2250vh8.4

D'Antonio C, Dudley T (1994) Biological invasions as agents of change on islands versus mainlands. In: Vitousek PM, Loope L, Andersen H (Eds) Islands: Biological diversity and ecosystem function. Springer-Verlag, Berlin, 103–121. https://doi.org/10.1007/978-3-642-78963-2_9

Day MD, Riding N (2019) Host specificity of *Puccinia spegazzinii* (Pucciniales: Pucciniaceae), a biological control agent for *Mikania micrantha* (Asteraceae) in Australia. Biocontrol Science and Technology 29: 19–27. https://doi.org/10.1080/09583157.2018.1520807

Denslow JS (2003) Weeds in paradise: thoughts on the invasibility of tropical islands. Annals of the Missouri Botanical Garden 1: 119–127. https://doi.org/10.2307/3298531

Director of National Parks (2014) Christmas Island National Park Management Plan 2014–2024. [ISBN: 978-0-9807460-3-7] https://www.environment.gov.au/system/files/resources/aa0c6d7f-d787-405b-955e-3bd9d1281ca7/files/christmas-island-national-park-management-plan-2014-2024_1.pdf

Dodd J, Reeves AW, Eldershaw V (2012) Overcoming weed eradication challenges in the Indian Ocean Territories. In: Eldershaw V (Ed.) Proceedings of the 18th Australasian Weeds Conference. Weed Society of Victoria, Melbourne, 263–266.

Fine PVA (2002) The invasibility of tropical forests by exotic plants. Journal of Tropical Ecology 18: 687–705. https://doi.org/10.1017/S0266467402002456

Fletcher CS, Westcott DW (2013) Dispersal and the design of effective management strategies for plant invasions: matching scales for success. Ecological Applications 23: 1881–1892. https://doi.org/10.1890/12-2059.1

Fletcher CS, Westcott DA, Murphy HT, Grice AT, Clarkson JR (2015) Managing breaches of containment and eradication of invasive plant populations. Journal of Applied Ecology 52: 59–68. https://doi.org/10.1111/1365-2664.12361

Flora of Australia (1993) Volume 50, Oceanic Islands 2. Australian Government Publishing Service, Canberra.

FAO [Food and Agriculture Organisation of the United Nations] (2011) Procedures for Post-Border Weed Risk Management. 2nd edn. Food and Agriculture Organisation of the United Nations, Rome.

Geoscience Australia (2014) Christmas Island vegetation and clearing map. Geoscience Australia, Australian Government, Canberra.

Gimeno I, Vilà M, Hulme PE (2006) Are islands more susceptible to plant invasion than continents? A test using *Oxalis pes-caprae* in the western Mediterranean. Journal of Biogeography 33: 1559–1565. https://doi.org/10.1111/j.1365-2699.2006.01525.x

Green PT, Hart R, Bin Jantan J, Metcalfe DJ, O'Dowd DJ, Lake PS (1999) Red crabs in rainforest on Christmas Island, Indian Ocean: No effect on the soil seed bank. Australian Journal of Ecology 24: 90–94. https://doi.org/10.1046/j.1442-9993.1999.00950.x

Green PT, Lake P, O'Dowd DJ (2004) Resistance of island rainforest to invasion by alien plants: Influence of microhabitat and herbivory on seedling performance. Biological Invasions 6: 1–9. https://doi.org/10.1023/B:BINV.0000010144.12808.cb

Grice AC, Clarkson JR, Friedel MH, Ferdinands K, Setterfield S (2010) Containment as a strategy for tackling contentious plants. In: Zydenbos SM (Ed.) Proceedings of the 17th Australasian Weeds Conference. New Zealand Plant Protection Society, Christchurch, 486–489.

Grice T (2009) Principles of containment and control. In: Clout MN, Williams PA (Eds) Invasive Species Management. Oxford University Press, Oxford, 61–76.

Grimes KG (2001) Karst features of Christmas Island (Indian Ocean). Helictite 37(2): 41–58.

Heger WT, van Andel T (2019) A social-ecological perspective on ecosystem vulnerability for the invasive creeper coralita (*Antigonon leptopus*) in the Caribbean: A review. Global Ecology and Conservation 18: e00605. https://doi.org/10.1016/j.gecco.2019.e00605

Higgins SI, Richardson, DM (1996) A review of models of alien plant spread. Ecological Modelling 87: 249–265. https://doi.org/10.1016/0304-3800(95)00022-4

Hulme PE (2006) Beyond control: Wider implications for the management of biological invasions. Journal of Applied Ecology 43: 835–847. https://doi.org/10.1111/j.1365-2664.2006.01227.x

Hulme PE (2020) Plant invasions in New Zealand: global lessons in prevention, eradication and control. Biological Invasions 22: 1539–1562. https://doi.org/10.1007/s10530-020-02224-6

James DJ, Green PT, Humphreys WF, Woinarski JC (2019) Endemic species of Christmas Island, Indian Ocean. Records of the Western Australian Museum 34(2): 55–114. https://doi.org/10.18195/issn.0312-3162.34(2).2019.055-114

Kumschick S, Bacher S, Bertolino S, Blackburn TM, Evans T, Roy HE, Smith K (2020) Appropriate uses of EICAT protocol, data and classifications. NeoBiota 62: 193–212. https://doi.org/10.3897/neobiota.62.51574

Lohr MT, Lohr CA, Keighery G, Long V (2016) The status and distribution of non-native plants on the gazetted and territorial islands off the north coast of Western Australia. Conservation Science Western Australia 10: 8. [online] https://www.dpaw.wa.gov.au/CSWAjournal

Lonsdale WM (1999) Global patterns of plant invasions and the concept of invasibility. Ecology 80: 1522–1536. https://doi.org/10.1890/0012-9658(1999)080[1522:GPOPIA]2.0.CO;2

Macanawai AR, Day MD, Adkins SW (2018) Seed biology of *Mikania micrantha* in Viti Levu, Fiji. Weed Research 58: 229–238. https://doi.org/10.1111/wre.12299

MacDiarmond SC, Pharo HJ (2003) Risk analysis: Assessment, management and communication. Revue Scientifique et Technique – Office International des Epizooties 22: 397–408. https://doi.org/10.20506/rst.22.2.1408

Moore JL, Runge, MC, Webber BL, Wilson, JRU (2011) Contain or eradicate? Optimising the goal of managing Australian acacia invasions in the face of uncertainty. Diversity and Distributions 17: 1047–1059. https://doi.org/10.1111/j.1472-4642.2011.00809.x

Myers JH, Savoie A, van Randen E (1998) Eradication and pest management. Annual Review of Entomology 43: 471–491. https://doi.org/10.1146/annurev.ento.43.1.471

Nathan R (2006) Long-distance dispersal of plants. Science 313: 786–788. https://doi.org/10.1126/science.1124975

Panetta FD (2007) Evaluation of weed eradication programs: containment and extirpation. Diversity and Distributions 13: 33–41. https://doi.org/10.1111/j.1472-4642.2012.00932.x

Panetta FD (2009) Weed eradication: an economic perspective. Invasive Plant Science and Management 2: 360–368. https://doi.org/10.1614/IPSM-09-003.1

Panetta FD (2012) Evaluating the performance of weed containment programmes. Diversity and Distributions 18: 1024–1032. https://doi.org/10.1111/j.1472-4642.2012.00932.x

Panetta FD (2015) Weed eradication feasibility: lessons of the 21st century. Weed Research 55: 226–238. https://doi.org/10.1111/wre.12136

Panetta FD, Cacho OJ (2012) Beyond fecundity control: which weeds are most containable? Journal of Applied Ecology 49: 311–321. https://doi.org/10.1111/j.1365-2664.2011.02105.x

Panetta FD, Cacho OJ (2014) Designing weed containment strategies: An approach based on feasibilities of eradication and containment. Diversity and Distributions 20: 555–566. https://doi.org/10.1111/ddi.12170

Panetta FD, Gooden B (2017) Managing for biodiversity: impact and action thresholds for invasive plants in natural ecosystems. NeoBiota 34: 53–66. https://doi.org/10.3897/neobiota.34.11821

Panetta FD, Timmins SM (2004) Evaluating the feasibility of eradication for terrestrial weed invasions. Plant Protection Quarterly 19: 5–11.

Parks Australia (2009) Christmas Island Weed Management Plan. Department of Environment, Canberra.

Parry HR, Sadler RJ, Kriticos DJ (2013) Practical guidelines for modelling post-entry spread in invasion ecology. NeoBiota 18: 41–66. https://doi.org/10.3897/neobiota.18.4305

Pheloung PC, Williams PA, Halloy SR (1999) A weed risk assessment model to use as a biosecurity tool evaluating plant introductions. Journal of Environmental Management 57: 239–251. https://doi.org/10.1006/jema.1999.0297

Reaser JK, Meyerson LA, Cronk Q, De Poorter MA, Eldrege LG, Green E, Kairo M, Latasi P, Mack RN, Mauremootoo J, O'Dowd D (2007) Ecological and socioeconomic impacts of invasive alien species in island ecosystems. Environmental Conservation 98–111. https://doi.org/10.1017/S0376892907003815

Rejmánek M, Pitcairn MJ (2002) When is eradication of exotic pest plants a realistic goal? In: Vietch CR, Clout MN (Eds) Turning the tide: The eradication of island invasives. IUCN (International Union for Conservation of Nature), Gland, 249–253.

Richardson DM, Pysek P, Rejmánek M, Barbour MG, Panetta FD, West CJ (2000) Naturalization and invasion of alien plants: concepts and definitions. Diversity and Distributions 6: 93–107. https://doi.org/10.1046/j.1472-4642.2000.00083.x

Roseleine D, Suzuki E (2012) Secondary succession at abandoned grazing sites, Pangandaran Nature Reserve, West Java, Indonesia. Tropics 21: 91–103. https://doi.org/10.3759/tropics.21.91

Roberts WL (2017) False precision: the ring of truth. Canadian Journal of Behavioural Science 49: 97–99. https://doi.org/10.1037/cbs0000070

Scott JK, Batchelor KL, Webber BL (2019) Long term monitoring of recruitment dynamics determines eradication feasibility for an introduced coastal weed. NeoBiota 50: 31–53. https://doi.org/10.3897/neobiota.50.35070

Sharov AA, Liebhold AM (1998) Bioeconomics of managing the spread of exotic pest species with barrier zones. Ecological Applications 8: 833–845. https://doi.org/10.1890/1051-0761(1998)008[0833:BOMTSO]2.0.CO;2

Simberloff D (2003) Eradication – preventing invasions at the outset. Weed Science 51: 247–253. https://doi.org/10.1614/0043-1745(2003)051[0247:EPIATO]2.0.CO;2

Strubbe D, White R, Edelaar P, Rahbek C, Shwartz A (2019) Advancing impact assessments of non-native species: strategies for strengthening the evidence-base. NeoBiota 51: 41–64. https://doi.org/10.3897/neobiota.51.35940

Swarbrick JT, Hart R (2000) Environmental weeds of Christmas Island and their management. Plant Protection Quarterly 16: 54–57.

Vanderhoeven S, Branquart E, Casaer J, D'hondt B, Hulme PE, Shwartz A, Strubbe D, Turbé A, Verreycken H, Adriaens T (2017) Beyond protocols: improving the reliability of expert-based risk analysis underpinning invasive species policies. Biological Invasions 19: 2507–2517. https://doi.org/10.1007/s10530-017-1434-0

Vieira DCM, Socolowski F, Takaki M (2010). Seed germination and seedling emergence of the invasive exotic species, *Clausena excavata*. Brazilian Journal of Biology 70: 1015–1020. https://doi.org/10.1590/S1519-69842010000500014

Vilà M, Tessier M, Suehs CM, Brundu G, Carta L, Galanidis A, Lambdon P, Manca M, Médail F, Moragues E, Traveset A, Troumbis AY, Hulme PE (2006) Local and regional assessment of the impacts of plant invaders on vegetation structure and soil properties of Mediterranean islands. Journal of Biogeography 33: 853–861. https://doi.org/10.1111/j.1365-2699.2005.01430.x

Virtue JG, Groves RH, Panetta FD (2001) Towards a system to determine the national significance of weeds in Australia. In: Groves RH, Panetta FD, Virtue JG (Eds) Weed Risk Assessment. CSIRO Publishing, Melbourne, 124–150.

Williams LK, Fergus AJ, Shaw JD, Terauds A, Kristiansen P, Wilson SC, Gosden JL, Ziegler K, Sindel BM (2019) Quantifying site and species factors to inform the feasibility of eradication of alien plants from Southern Ocean Islands: *Stellaria media* on Macquarie Island. Biological Invasions 21: 993–1005. https://doi.org/10.1007/s10530-018-1880-3

Wilson JR, Panetta FD, Lindgren CJ (2017) Detecting and responding to alien plant incursions. Cambridge University Press, Cambridge, 265 pp. https://doi.org/10.1017/CBO9781316155318

Wilson JRU, Gairifo C, Gibson MR, Arianoutsou M, Bakar BB, Baret S, Celesti-Grapow L, DiTomaso JM, Dufour-Dror J-M, Kueffer C, Kull CA, Hoffmann, JH, Impson FAC, Loope LL, Marchante E, Marchante H, Moore JL, Murphy, DJ, Tassin J, Witt A, Zenni RD, Richardson DM (2011) Risk assessment, eradication, and biological control: global efforts to limit Australian acacia invasions. Diversity and Distributions 17: 1030–1046. https://doi.org/10.1111/j.1472-4642.2011.00815.x